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# WATER QUALITY ASSESSMENT OF LISGAR LAKE, TILLSONBURG, ONTARIO

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# WATER QUALITY ASSESSMENT OF LISGAR LAKE, TILLSONBURG, ONTARIO

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December 1991

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## PREFACE

Lisgar Lake was one of 11 southwestern Ontario lakes investigated by the Ministry of the Environment as part of the Inland Lakes Blue-green algae control program (1987-1989). The program was designed to monitor a selection of hardwater inland lakes with, or with the potential for, nuisance blue-green algae blooms. Lakes were chosen in response to complaints regarding surface scums and algal blooms, received from organized public associations or Conservation Authorities. Based on these surveys, several lakes were assessed as having a good potential to respond to experimental treatment programs.

At this time, no further work is planned for Lisgar Lake. This report was prepared to ensure that the water quality information is made available to those agencies and individuals expressing an interest in the findings.

## ABSTRACT

Lisgar Lake is a 6.8 ha hardwater lake with a maximum depth of 6.8 m. Centrally located in the town of Tillsonburg, it provides important aesthetic and recreational benefits to the public in a region where lakes of recreational size are rare. This report summarizes the water quality of Lisgar Lake during the ice-free season of 1987.

Eutrophic Lisgar Lake experienced peak phytoplankton densities in late summer. Chlorophyll a levels were correlated to algal biovolumes and reached concentrations as high as 62 $\mu$g/l, with an annual mean of 19.6 $\mu$g/l. As the season progressed, the algal assemblage became increasingly dominated by blue-green algae which comprised 60 % of the total algal biovolume by late August. Water clarity, as measured by Secchi disc, varied from 4.0 m in the spring to less than 1.0 m by late summer. Mean annual TN:TP ratios were 173:1, but fell to less than 5:1 during the blue-green bloom episodes.



Gemza, A.F. 1991. Water quality assessment of Lisgar Lake, Tillsonburg, Ontario. Ontario Ministry of the Environment, Limnology Section, Aquatic Plant Unit. pp 42.

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## INTRODUCTION

Lisgar Lake, located within the town limits of Tillsonburg, Ontario (Lat. 42° 55', Long. 80° 51'), is a former rock quarry allowed to fill with water in the 1960s, and is situated on glacial till of limestone and shale. Several natural springs feed the lake. Measurements first taken in 1973 by the Ministry of Natural Resources show that the groundwaters flow at a rate of 4.0 to 5.0 cubic feet per second. Lisgar Lake was originally two quarry ponds, that were joined to form the present lake when a concrete overflow sluice was constructed raising water levels.

With a surface area of 6.8 hectares and a maximum depth of 6.8 meters, Lisgar Lake was one of the smaller lakes monitored in the region. It has a mean depth of 3.9 m and a storage capacity as a man-made reservoir of $0.339 \times 10^6$ $m^3$.

By 1987, land use on the south side was predominantly urban, and roads now surround 75% of the lake, with the required construction of stormwater drains. Several culverts which spill into Lisgar Lake are visible around its perimeter. Flow out of the lake is surface drawn into a simple concrete overflow sluice, which feeds a municipal drain that empties into a tributary of Otter Creek.

After the gravel pits were decommissioned, they were used by townspeople for recreational swimming. Swimming is now discouraged since the stormwater drains may carry a high bacterial load from surface runoff. Sport fishermen and local rowing enthusiasts are regular users of the lake. A public park remains on the north shore of the lake, and an abundant variety of hardwood tree species like birch, maples, beech, oak, and many types of shrubs, line the shores and provide good bank stabilization.

In the late 1960s and early 1970s, local user groups began lake reclamation efforts. In 1968, the lake was poisoned with rotenone to eradicate populations of coarse fish (carp, goldfish and suckers), to prepare the lake for major fish stocking efforts. Commencing

with stockings of rainbow and brook trout in the fall of 1968, over 20,000 fish were stocked by 1973 (MNR, pers. comm.). Stocking was carried on into the mid 1970s. A review of local newspaper articles during this time period shows that regular fishing derbies were major town events.

The current fish community may well contain remnants of these early plantings. As well, there have been regular sightings of smallmouth bass and panfish, and several redds.

Lisgar Lake was treated with herbicides ("Reglone A"), (20% diquat), several times in the 1970s to control nuisance aquatic plants of predominantly Typha sp. and a variety of Potomageton species. The early season macrophyte community was dominated in 1987 by P. crispus (curly leaf pondweed) in May and June, and succeeded by predominantly Myriophyllum sp. by July. Dense macrophyte communities appear limited to the north shore of the lake.

Herbicide treatment and fish stocking were not known to take place during 1987.

In response to reports to the Ministry of the Environment that Lisgar Lake experienced mid to late-summer blue-green algal blooms and lake-wide odours, an assessment of its water quality status was conducted.

## METHODS

Water samples and oxygen/temperature profiles were collected approximately bi-weekly between May 11 and September 2 during 1987. Samples for water chemistry were collected at one site, 50 meters west of the spillway at the deepest point (6.8 m) of the lake (Figure 1).

Two narrow mouthed (2.5 cm) 1-litre glass bottles were lowered and raised through the euphotic zone, (twice the Secchi disc depth), to a maximum of one meter off bottom

collecting a composite sample of the water column. The rate of descent and ascent was timed so that the bottles were completely filled just as they reached the surface. The Secchi disc is a 20 cm. diameter plate painted in alternating black and white quarters, used to determine water clarity. The disc was lowered on the shaded side of the boat until it disappeared, then raised again until it reappeared. The average of these two depths was termed the Secchi disc depth.

A 500 ml subsample was withdrawn into a separate bottle for chlorophyll analysis, which was stabilized with 2 ml of a 2% $MgCO_3$ suspension. A second 500 ml subsample for metals analyses was preserved with 2 ml of concentrated nitric acid to prevent metals precipitation. A third subsample was preserved with Lugol's iodine for phytoplankton analyses and a fourth sample for nutrient chemistry was unpreserved. Samples were kept cool and delivered to the Ontario Ministry of the Environment's laboratory the same day for analyses. Samples were refrigerated and analyzed within 48 hours of collection following standard procedures (MOE 1981).

A second set of samples was drawn from 1 meter off the bottom of the lake (1 MOB), using a 6 litre capacity PVC Van-Dorn bottle. 500 ml samples were withdrawn from the bottle for nutrient chemistry and metals analyses.

Oxygen and temperature measurements were taken at one meter intervals from the surface of the lake to the bottom, using a YSI 58 dissolved oxygen and temperature meter. The meter was calibrated against a standard hand held mercury thermometer for temperature accuracy, and calibrated against water samples of known oxygen concentration, determined by the micro-Azide modification of the Winkler technique. In addition, 60 ml Winkler samples from the lake surface and 1 MOB were taken to confirm the oxygen profiles.

Zooplankton samples were collected using a modified, metered Clarke-Bumpus net, with 80 $\mu$m mesh and a 15.5 cm mouth. The collection net was raised through a vertical

column from 1 MOB to the surface with the net attached to collect the sample (A), and with the net removed (B), to determine net flow restriction. Efficiency was determined by the ratio A/B from the meter values of each haul.

## RESULTS

### Oxygen and Temperature

As Lisgar Lake is relatively well protected by high, well forested banks, wind mixing is gentle and summer surface temperatures rose as high as 28° C. by late July. The lake stratified quickly, and by the end of May, a strong thermocline had formed at 3 meters (Figure 2). Hypolimnetic temperatures rose to 16.3 °C in August, as the thermocline progressively eroded, allowing partial turnover to occur in early September. Rising bottom water temperatures are likely the result of warming groundwater flow as the season progressed, and light and heat penetration through the shallow epilimnion.

Surface waters were well oxygenated ranging between 9.0 to 16.2 mg/L . Metalimnetic maxima were observed throughout the early part of the summer (Figure 3 ), with levels as much as 4.0 mg/L higher than in the epilimnion. Water at 6 m was already anoxic at the first monitoring in early May. Hypolimnetic anoxia was, however, limited to 5 m through June and July. Oxygen levels fell to 2.5 mg/L in water as shallow as 3.0 m for a short period in late August (Figure 3). At this time, $H_2S$ odours were distinct in samples retrieved from 1 MOB. Oxygen and temperature profiles for 1987 are presented in Figure 2 and Figure 3.

### Secchi Depth and Turbidity

Secchi disc measurements ranged between 4.0 m in the early spring to 0.8 m during late summer phytoplankton blooms (Figure 4). An annual mean of 2.2 m compared favourably to water clarity of other lakes in this region (unpubl. data).

Euphotic zone turbidity varied little seasonally, with an annual mean of 4.1 Formazin turbidity units (FTU). Sharply higher levels as high as 45 FTU, however, were observed at 1 MOB throughout August (Figure 5 ).

**Phosphorus**

During most of the summer, 1 MOB and euphotic zone total phosphorus (TP) levels had similar trends. Euphotic zone TP averaged 30 $\mu$g/L and was higher in bottom waters. TP peaked in September following a rapid increase in 1 MOB levels, which reached concentrations of 280 $\mu$g/L (Figure 6).

Phosphate ($PO_4^{-3}$) trends were more variable. 1 MOB levels exhibited spring and late summer peaks to 28 $\mu$g/L with a mean of 10 $\mu$g/L. Euphotic zone $PO_4^{-3}$ peaked in mid summer at 5.5 $\mu$g/L, and was relatively stable throughout the rest of the year, with a mean of 2 $\mu$g/L (Figure 7).

**Nitrogen**

Seasonal euphotic zone Kjeldahl (TKN) nitrogen (Figure 9 ) and ammonium (Figure 8) trends were similar to those of $PO_4^{-3}$, with stable levels throughout most of the season. 1 MOB concentrations peaked in September, with a high of 1.72 mg/L for ammonium and 2.75 mg/L for TKN (Figure 9).

Total nitrates ($NO_3^-$ + $NO_{2-}$) in the euphotic zone and at 1 MOB displayed identical trends of spring concentration peaks, gradually falling to within minimum detection levels by the end of the growing season (Figure 10). Mean annual values are presented in Table 1. As a result, the TN:TP ratio displayed a strong spring peak, falling as the season progressed (Figure 15). Lisgar Lake had the highest annual TN:TP mean (173:1) of all the lakes monitored (unpubl. data).

## Chlorophyll a and Phytoplankton

Mean annual total chlorophyll a measured 19.6 $\mu$g/L. Seasonal fluctuations were tied to phytoplankton biovolumes (Figure 15). As expected, correlation between the two was positive ($R^2$ = .55). Chlorophyll a and phytoplankton levels were greatest in the late summer (Figure 16), indicative of phytoplankton blooms.

Phytoplankton levels through early July were below 2.0 $mm^3/L$, but biovolumes increased to 14.9 $mm^3/L$ by the end of August (Figure 16). The early season community was dominated by cryptophytes, however chlorophytes, predominantly Gloecystis sp. and, after July 7, Coelastrum sp. and Ceratium sp. (Dinophyceae), were dominant throughout the majority of the summer. Blue-green algae (Cyanophyceae), particularly Aphanizomenon sp. and Anabaena sp., comprised almost 60% of the algal assemblage by late summer (Figure 17). They were evident on the surface of the lake as floating scums. Diatoms (Bacillariophyceae) and chrysophytes were minor constituents of the phytoplankton.

## Zooplankton

The zooplankton were comprised of mainly three Daphnia sp., and copepods. Few other cladocerans were observed in 1987 (Figure 19). Zooplankton densities were highest in the spring. The population reached its climax (1200 $mg/m^3$) in late June. By early July, densities had fallen to approximately 200 $mg/m^3$, where they remained for the rest of the season (Figure 18). The Daphnia species were dominated in the early season by D. ambigua, D. parvula and D. pulex. D. pulex comprised 80% of the population during the June 23rd peak (Figure 18). For the remainder of the season, D. rosea was the dominant daphnid.

**Metals**

Euphotic zone iron (Fe) and manganese (Mn) levels showed mid-summer maxima, 0.34 mg/L and 0.86 mg/L respectively, while magnesium (Mg) levels gradually fell through the season from a spring peak of 14.8 mg/L (Figure 11). Annual euphotic zone means are presented in Table 1.

1 MOB seasonal trends showed stable levels predominating for the majority of the season, with strong late summer peaks (Figure 12). Late summer Fe levels exceeded 3.0 mg/L, Mg concentrations were over 15.0 mg/L and Mn levels peaked at 2.2 mg/L. High concentrations are most likely related to anoxic conditions in the hypolimnion. In the absence of oxygen, these metals become soluble.

**Conductivity, pH**

Mean annual conductivity of 459 $\mu$MHOs/cm was among the highest recorded in the region. The seasonal trend in the euphotic zone and at 1 MOB was of decreasing conductivity with time (Figure 13).

pH showed limited seasonal variation in response to photosynthetic activity, with a seasonal mean of 8.2 (Standard deviation 0.13).

**Alkalinity, Calcium and Hardness**

Euphotic zone alkalinity averaged 150.9 mg/L during 1987. The concentration showed a distinct drop in late summer due to increased algal production. During photosynthesis, levels fall as algal biovolumes increase. Calcium trends were similar (Figure 14). Alkalinity, calcium and hardness annual means were typical of well buffered hardwater lakes (Table 1).

## DISCUSSION

Shading from the densely treed shoreline appeared to reduce aquatic plant biomass and diversity along the south shore. Nuisance macrophyte and late summer blue-green densities were observed in areas where direct sunlight was most prevalent. Jorga et al. (1982) observed that tree stands along certain shores not only lowered total aquatic biomass, but reduced species diversity as well.

Poor late summer water clarity, as measured by Secchi disc, was primarily governed by phytoplankton densities ($r=0.63$) and the appearance of floating blue-green algae in late July and August. Euphotic zone turbidity remained constant through the year and could not be related to the reduction in Secchi disc visibility. Increased August turbidity at 1 MOB was most likely a result of settling algal cells, marl shaken loose from settling macrophytes and other decomposing plant matter suspended in the hypolimnion.

Although the input of groundwater nutrient contributions are unknown, increased late summer nutrient and metal concentrations at 1 MOB are likely a result of reducing (anoxic) conditions in the hypolimnion combined with the impact of settling algal remains, whose cell contents (nutrients) would be released upon decomposition. Oxygen is normally depleted as a result of biological respiration and the bacterial decomposition of organic matter (aquatic plant and algal debris), which settles on the lake bottom. It is well known that, in the absence of oxygen, the divalent (soluble) state of such micronutrients as iron and manganese predominates and, as a result, concentrations of these elements increased. This series of events takes place gradually through the summer, as bottom sediments become progressively reduced and nutrients and metals solubilize.

As plant biomass increases during the growing season, the gradual decrease in conductivity, alkalinity, DIC and calcium concentrations reflect plant and algal demand for these nutrients. As the demand for $CO_2$ and $HCO_3^-$ becomes excessive, the plants are forced to utilize carbon sources in solution. As a result, pH increases and $CaCO_3$

precipitates, as marl on plant surfaces and alkalinity and DIC levels fall.

Euphotic zone manganese concentrations rose dramatically in July to 0.86 mg/L. Levels greater than 1.0 mg/L may be inhibitory to blue-green and green algae, but should not be a concern at this concentration (Shapiro and Glass 1975). The source of the manganese is most likely a release from the massive die-back of Potomageton crispus (curlyleaf pondweed) at this time. P. crispus is a seasonal plant and is normally present only during the spring. Decomposing plants release their nutrients as they senesce (Landers 1982), and the mid summer epilimnetic spike of phosphate at the same time supports this observation. Manganese is also a component of fireworks, fertilizers and paints, and a limited amount may have entered the lake from the stormwater runoff of a nearby stadium and surrounding urban areas, following early July fireworks displays and activities. Magnesium is an essential element in plant metabolism, and is identified as a phosphate enzyme activator. As a result, euphotic zone levels decline during the summer as algal numbers, and therefore, demands for phosphorus, increase. Generally associated with calcium, it is usually present at about one third the calcium concentration in all natural waters. When Mg exceeds this amount, an industrial source is usually suspect. This is apparently not the case in Lisgar Lake.

Phosphorus has long been recognized as the limiting nutrient to algal production, and, through federal legislation, much emphasis has been placed on its control in detergents, and in its reduction from sewage treatment plant effluents. Phosphorus is present in fertilizers and may be entering the lake through runoff and leaching into groundwater. However, it appears that the bottom sediments are the primary source cycling phosphorus in Lisgar Lake. Under late summer oxygen deficient conditions, phosphorus was released to the water column. Iron, in particular, plays an important role in this type of release and the subsequent changes in lake trophic state.

When oxygen is depleted in the hypolimnion, and redox levels become sufficiently low, iron is reduced to the ferrous state and iron-phosphorus complexes break down, resulting

in phosphorus release from nutrient rich sediments (Syers 1973). The sediments, which act as a nutrient trap during oxidizing conditions, released high levels of phosphorus as well as associated metals into the hypolimnion. Under oxidized conditions, these metals bound with phosphorus. Under anaerobic conditions, microbial decomposition of organic matter and biochemical reduction of nitrate also took place, and hydrogen sulphide and ammonia levels in the hypolimnion rose as well. In the epilimnion, decreasing nitrate concentrations were most likely a result of algal uptake.

Vertical transport of nutrients (phosphorus, nitrogen) into the euphotic zone occurred primarily by wind mixing, resulting in higher euphotic zone concentrations. While limited diffusion through the thermocline is possible, euphotic zone increases were not observed until the depth of the thermocline was lowered in August.

As nutrients became available to the phytoplankton, blooms occurred and water quality and clarity were reduced. As described by Smith (1983), blue-green algae usually predominate when the TN:TP ratio falls below 29:1 and nitrate levels are low. Blue-green algae are able to fix atmospheric nitrogen and outcompete more desirable algal species, and generally, under these conditions, form surface scums. McQueen and Lean (1987) also proposed that, in waters with temperatures exceeding 21 °C, the likelihood of a blue-green bloom was greater. This was clearly the case in 1987.

Water clarity during May and June is likely the result of algal grazing by the large numbers of Daphnia sp. present at that time. Early season chlorophyll levels suggest that algal production was taking place in response to higher spring phosphate concentrations.

Decreasing densities of green algae (Chlorophyceae), prior to the peak of zooplankton abundance, are likely the result of selective zooplankton grazing. There appears to be a negative correlation between zooplankton numbers and phytoplankton biovolume.

Many recent studies suggest that food web dynamics significantly impact on water quality

(Gliwicz 1990). The Lisgar lake zooplankton community shifts, from one dominated by D. pulex in the early season, to almost exclusively, D. rosea. Many zooplankton are seasonal species, but the continued presence of D. pulex, even in late summer, points to size selective predation, since late summer daphnid species were smaller. Although measurements were variable, mean body lengths were observed to be up to 200 $\mu$m shorter in August.

The shift in zooplankton numbers and size, from large to smaller bodied specimens, may be a result of selective grazing on large bodied zooplankters by young-of-the-year fish. The abundance of a variety of fish and good breeding habitat suggest that this is a possibility. The senescence of the plant community may also have removed critical habitat that protected the zooplankton from predation during early summer (Crowder and Cooper 1982).

Both nutrient and food web dynamics play an important role in Lisgar Lake, but it cannot be determined from this limited assessment which of the two factors govern blue-green blooms in the lake more directly.

## CONCLUSIONS AND RECOMMENDATIONS

The treed shoreline should be maintained and lake shading increased with evergreen ornamental shrubs, because leaved tree varieties overhanging lakes input organic matter and add to winter oxygen depletion.

Higher densities of large piscivores (e.g., bass and pike) may reduce zooplanktivore numbers and result in greater zooplankton densities, in turn achieving lower phytoplankton biovolumes. Filter feeding zooplankton, however, appear to be unable to crop large filamentous blue-green algae because of their large size. As such, food web manipulation in Lisgar Lake would in all likelihood reduce total phytoplankton biovolume, but not the late summer blue-green blooms and their resulting surface scums.

Eliminating excessive nutrient release by oxidizing bottom waters and sediments is a more promising option. Euphotic zone nutrient levels in Lisgar Lake were closely tied to bottom water concentrations, and when mixed into the epilimnion, phytoplankton blooms formed. Hypolimnetic nutrient dynamics were governed by oxygen depletion. Bottom waters could be oxygenated by aeration, or by mechanical lake destratification, or with sediment oxidation by chemical addition.

Aeration, the bubbling of air to circulate the water mass, has had variable success. A variety of methods have been developed to eliminate stagnant bottom waters and the resulting oxygen depletion and sediment nutrient release.

In many cases, however, aeration has recirculated nutrients with gentle mixing and allowed blue-green algae to continue to dominate the phytoplankton (Lorenzen 1975), and even increased overall biovolumes. Aeration would likely prove beneficial to Lisgar Lake, but care must be exercised in the selection of the system, ensuring proper location and size.

The application of chemical oxidizing agents, specifically calcium nitrate $Ca(NO_3)^2$, holds some promise in the control of blue-green algae blooms. Added to bottom sediments in liquid or powder form, they oxidize the organic matter, alter nitrate levels, increase oxygen and release $CO_2$ by microbial action. Thus, $Ca(NO_3)^2$ may prevent phosphorus release and algal blooms (Ripl 1986). While still experimental, properly timed applications may provide good bloom control.

## ACKNOWLEDGEMENTS

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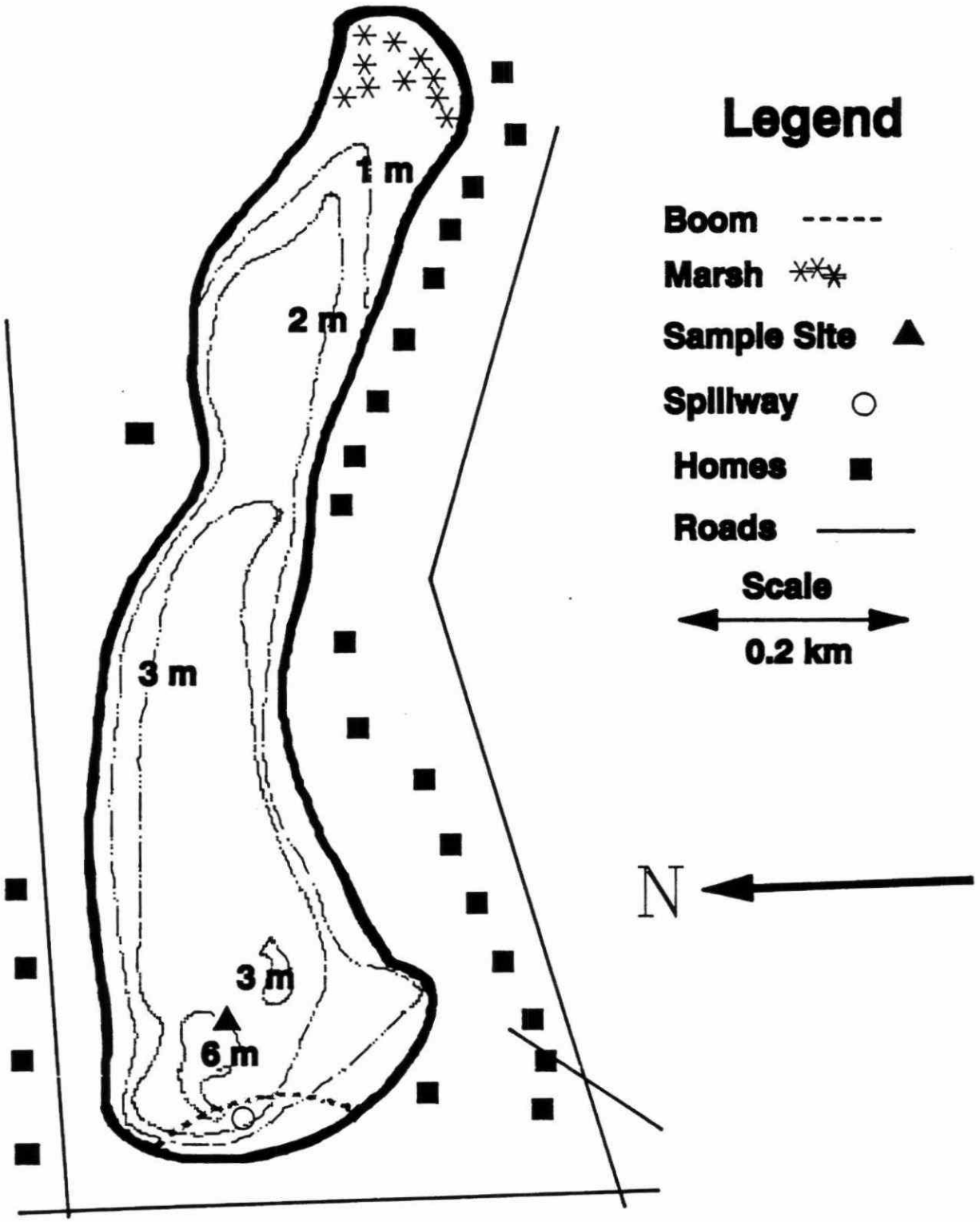


Figure 1: Bathymetric Map of Lisgar Lake.

Figure 2: Temperature profiles (°C.) between May 11 and Sept 2/1987 at Lisgar Lake.

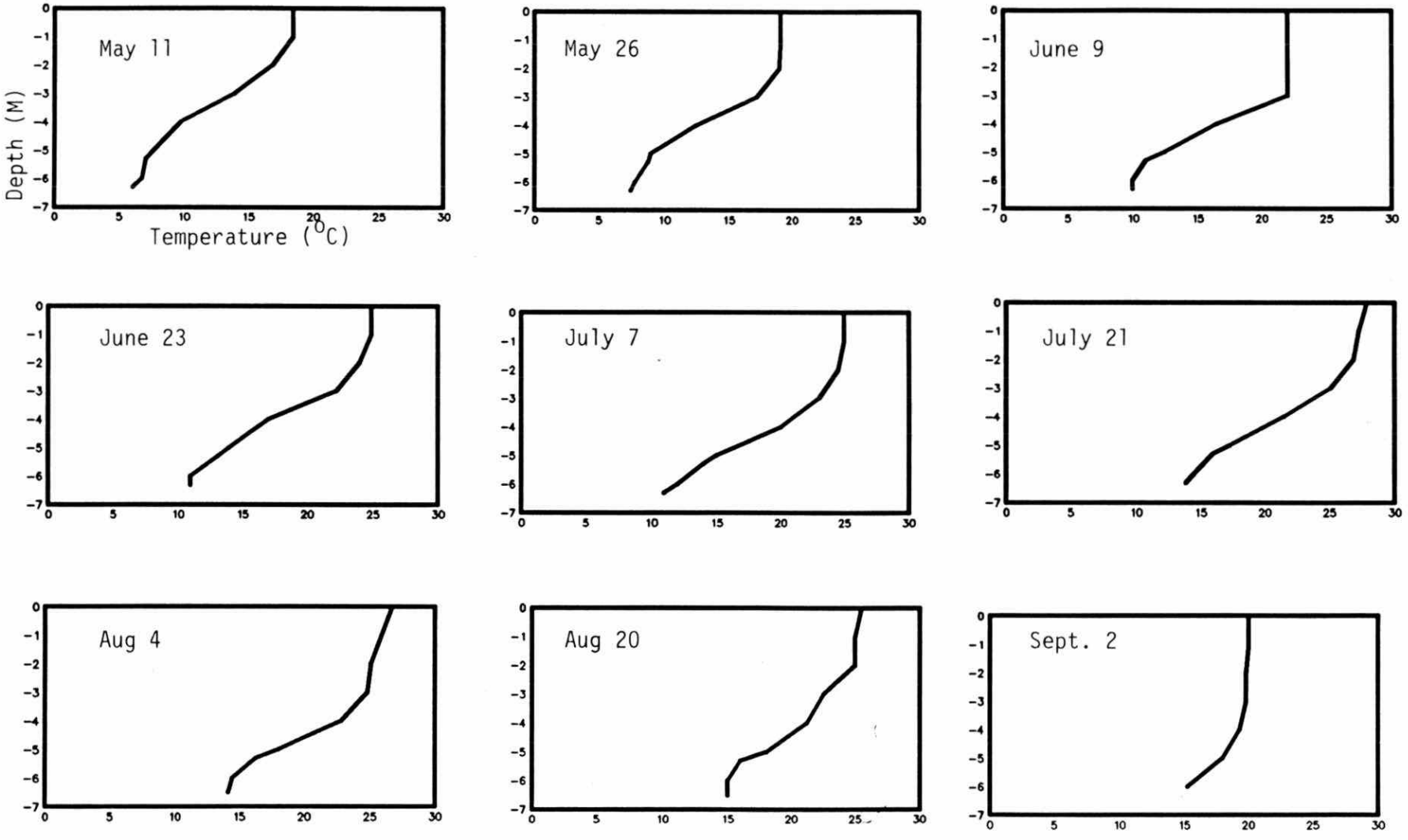

Figure 3: Dissolved oxygen profiles (mg/L) between May 11 and Sept 2/1987 at Lisgar Lake .

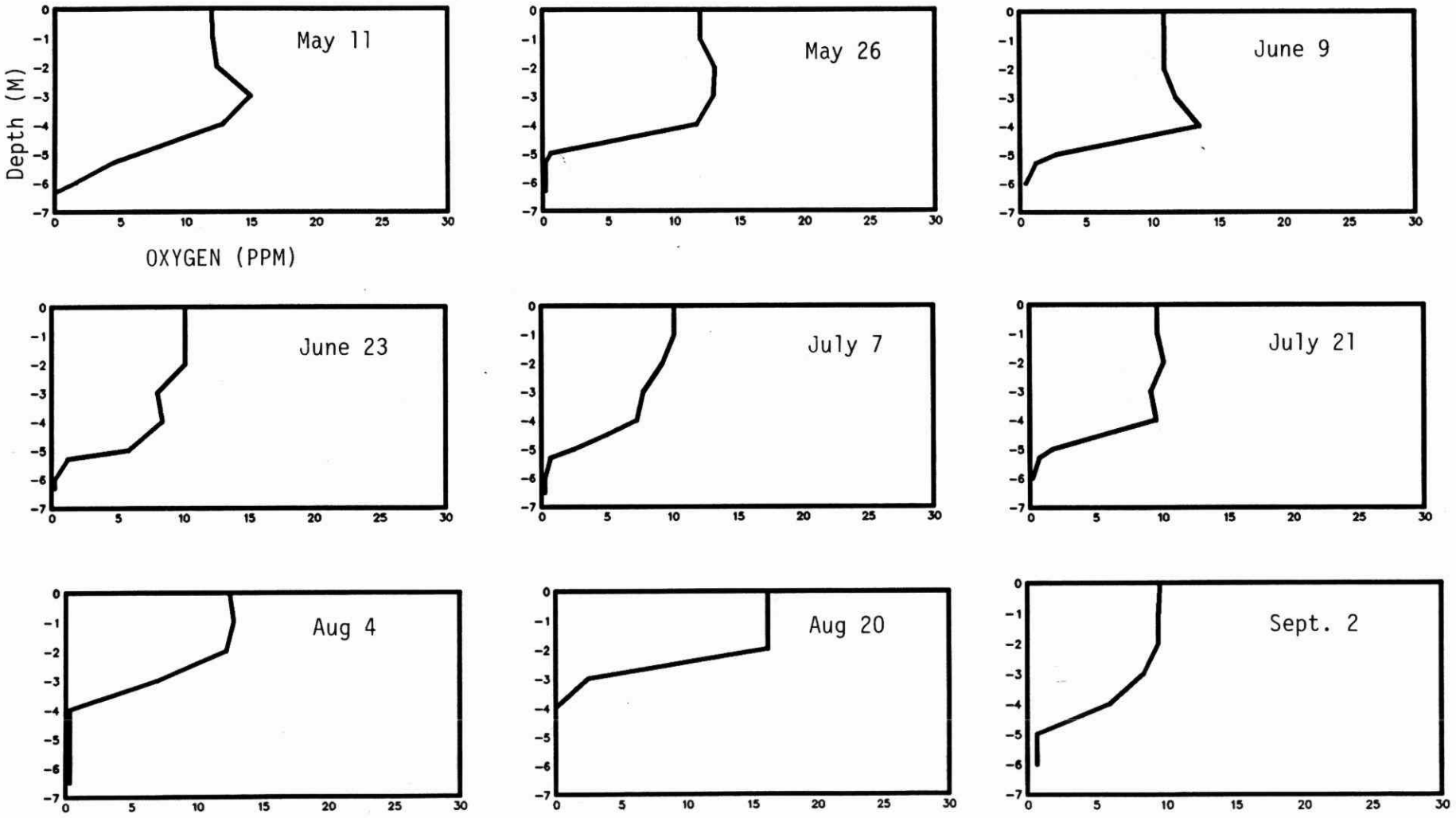

Figure 4: 1987 Secchi disc (m) visibility in Lisgar Lake, between May 11 and Sept 2/1987 .

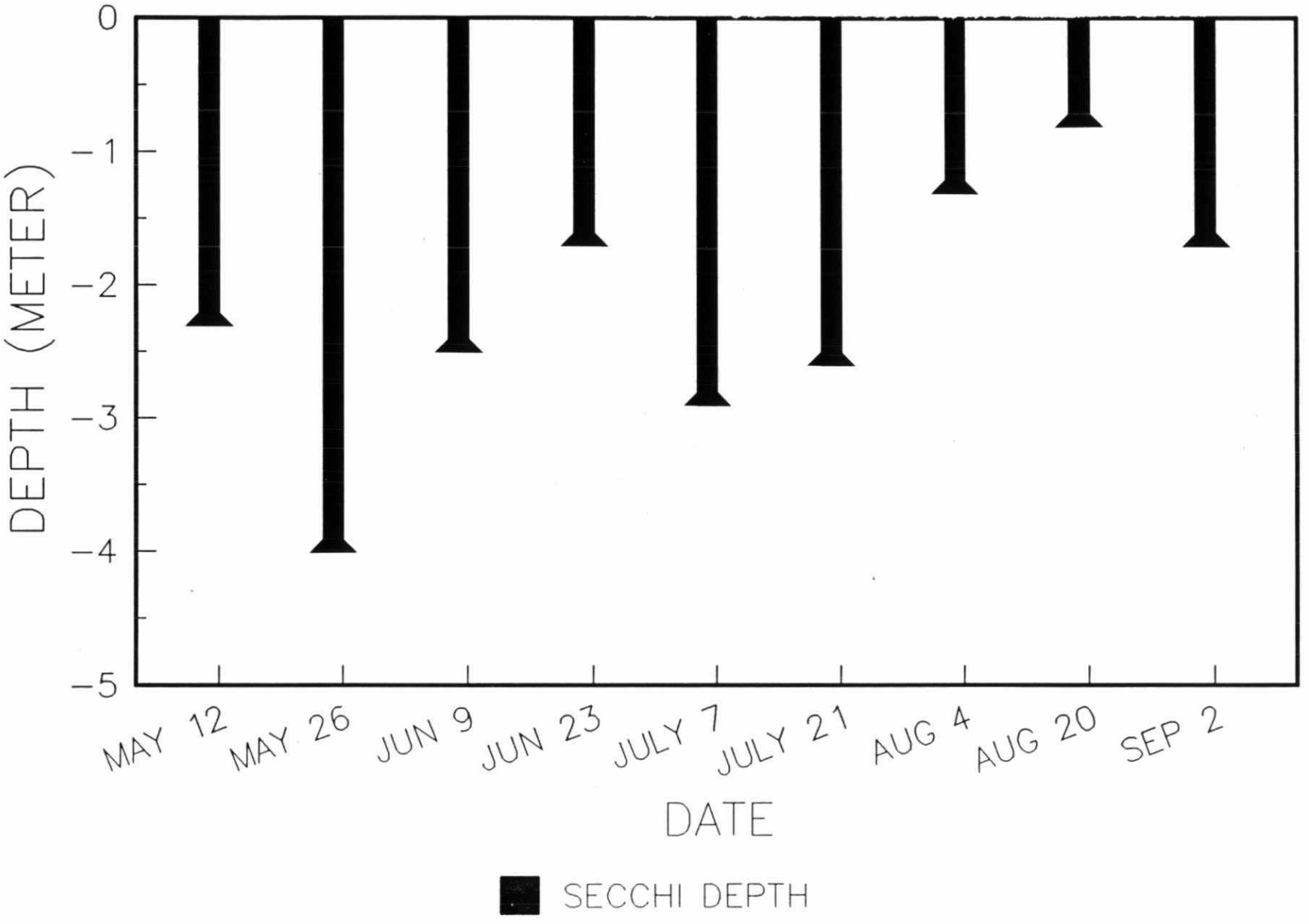

Figure 5: 1987 seasonal turbidity in the euphotic zone composite (comp) and 1 meter off bottom (1MOB)

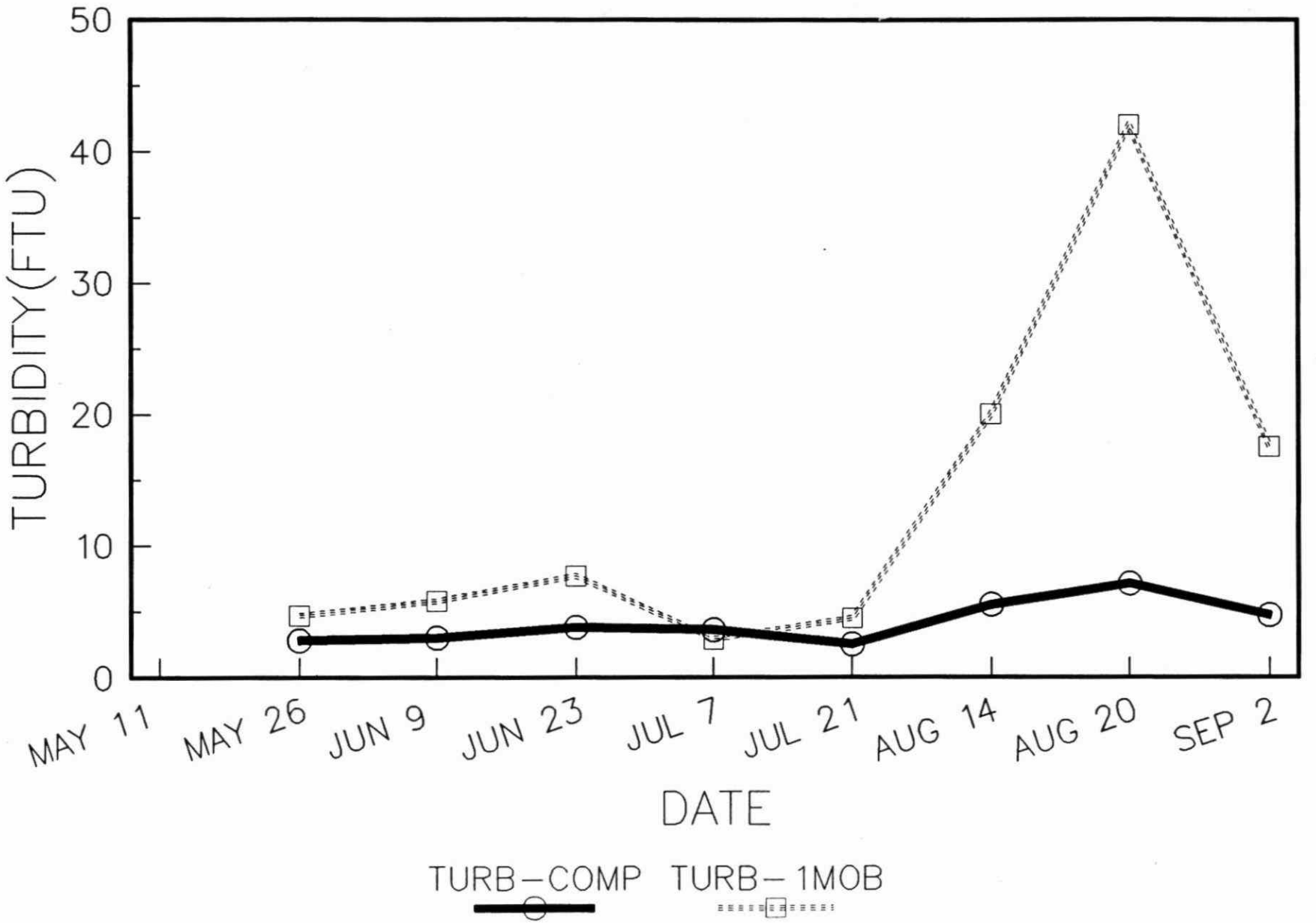

Figure 6: 1987 seasonal total phosphorus conentrations in the euphotic zone (comp) and at 1 MOB .

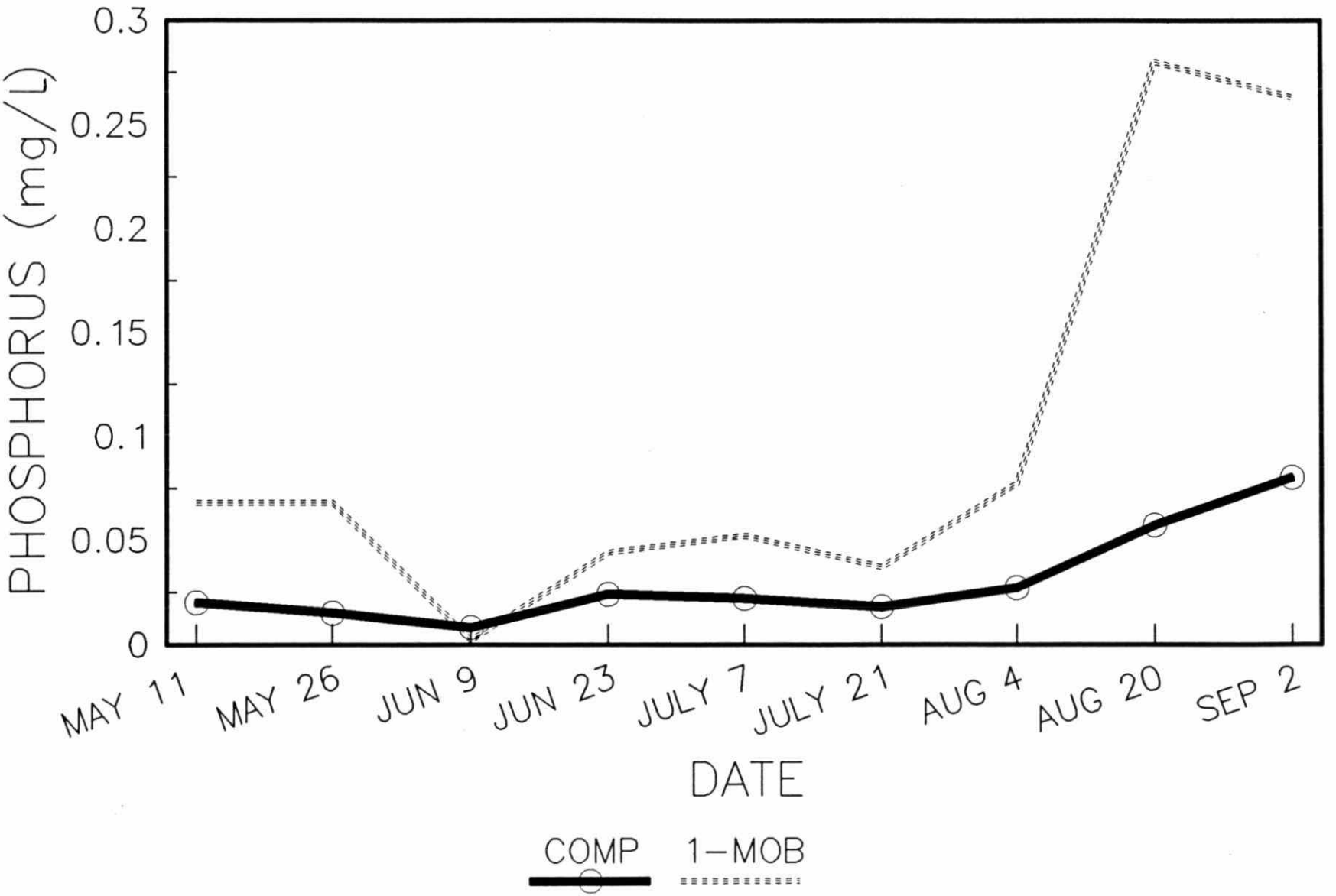

Figure 7: 1987 seasonal soluble reactive phosphate (SRP) concentrations in the euphotic zone (comp) and at 1 MOB.

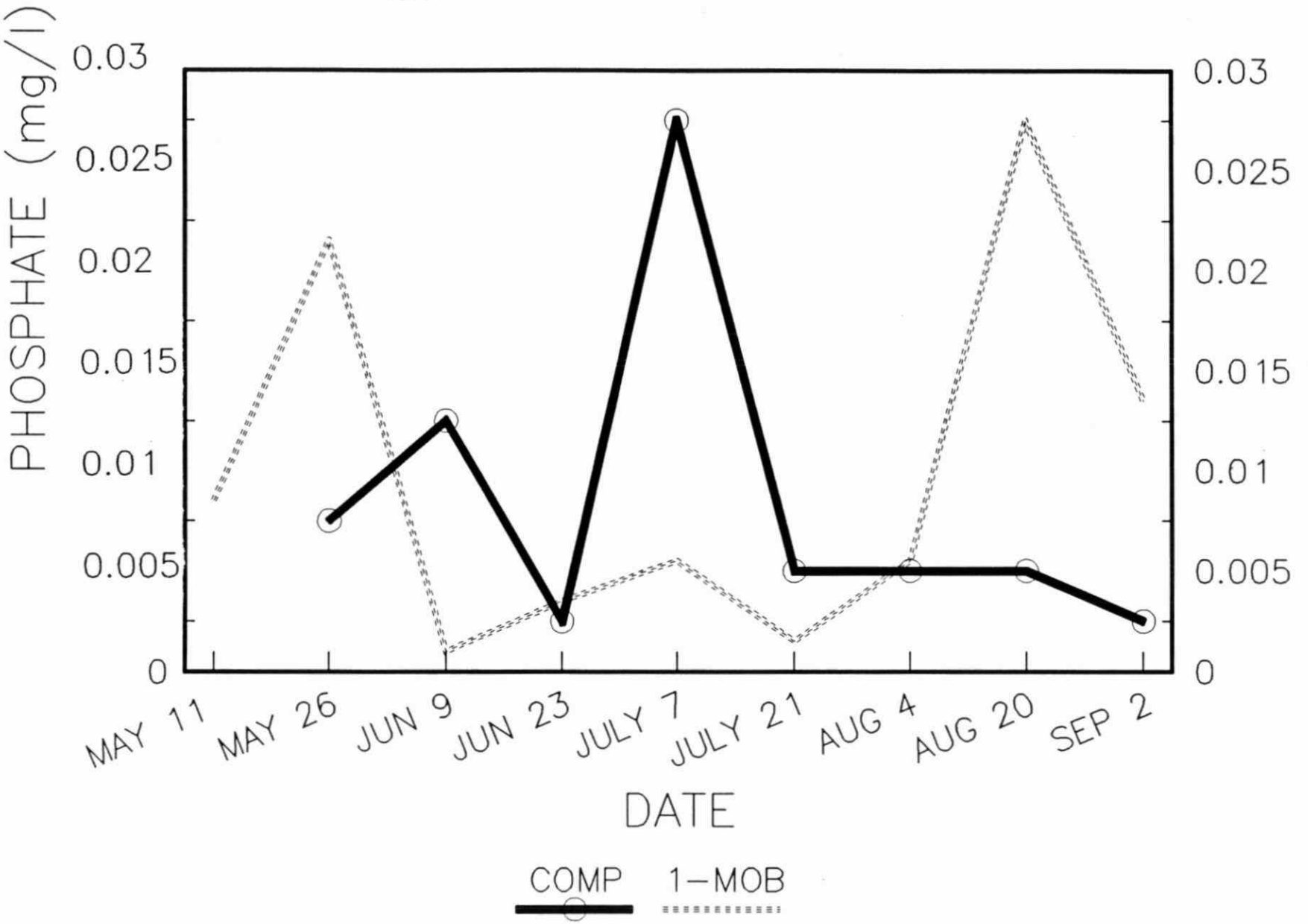

Figure 8: 1987 seasonal ammonium levels in the euphotic zone (comp) and at 1 MOB.

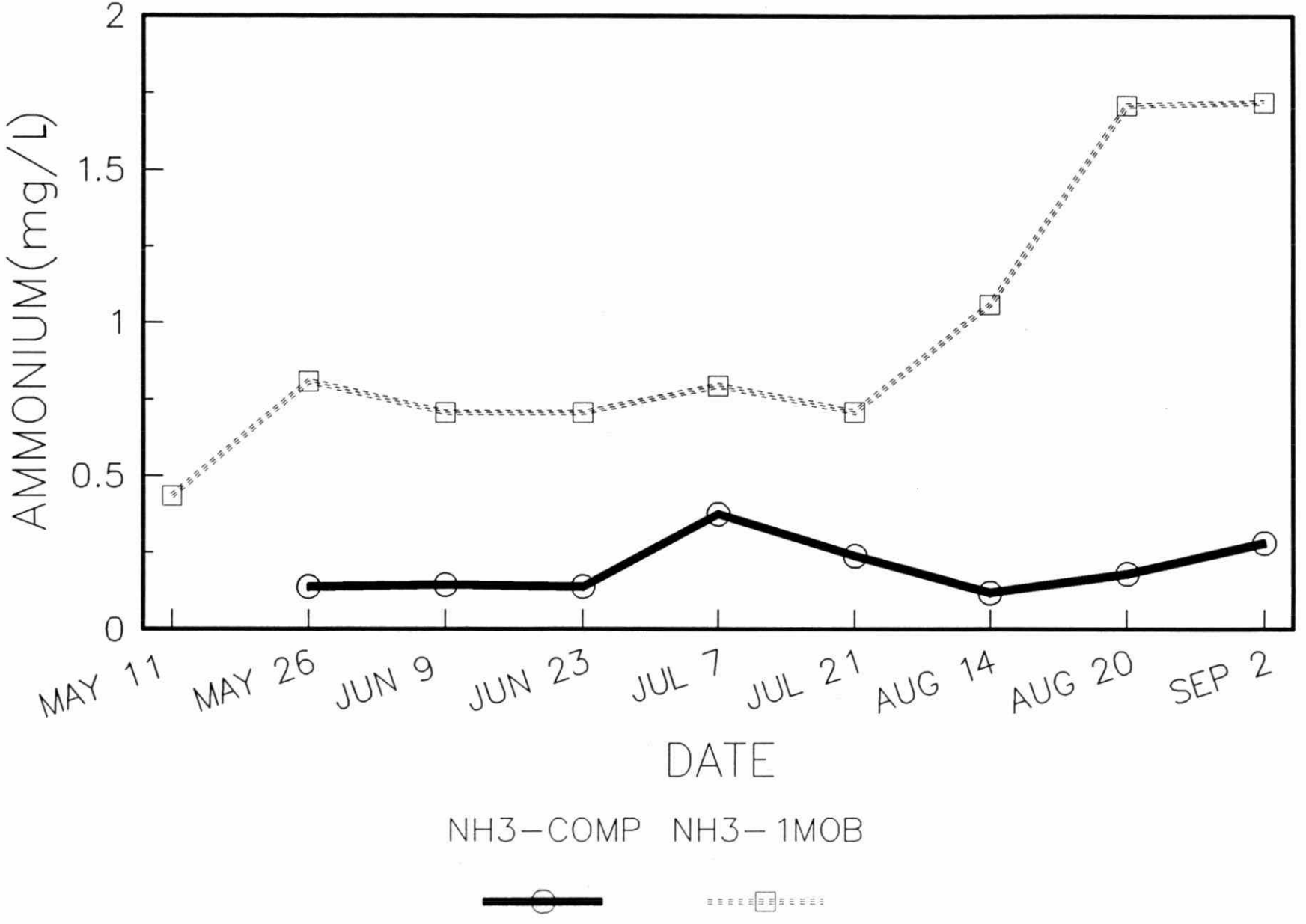

Figure 9: 1987 seasonal total Kjeldahl nitrogen (TKN) levels in the euphotic zone (comp) and at 1 MOB.

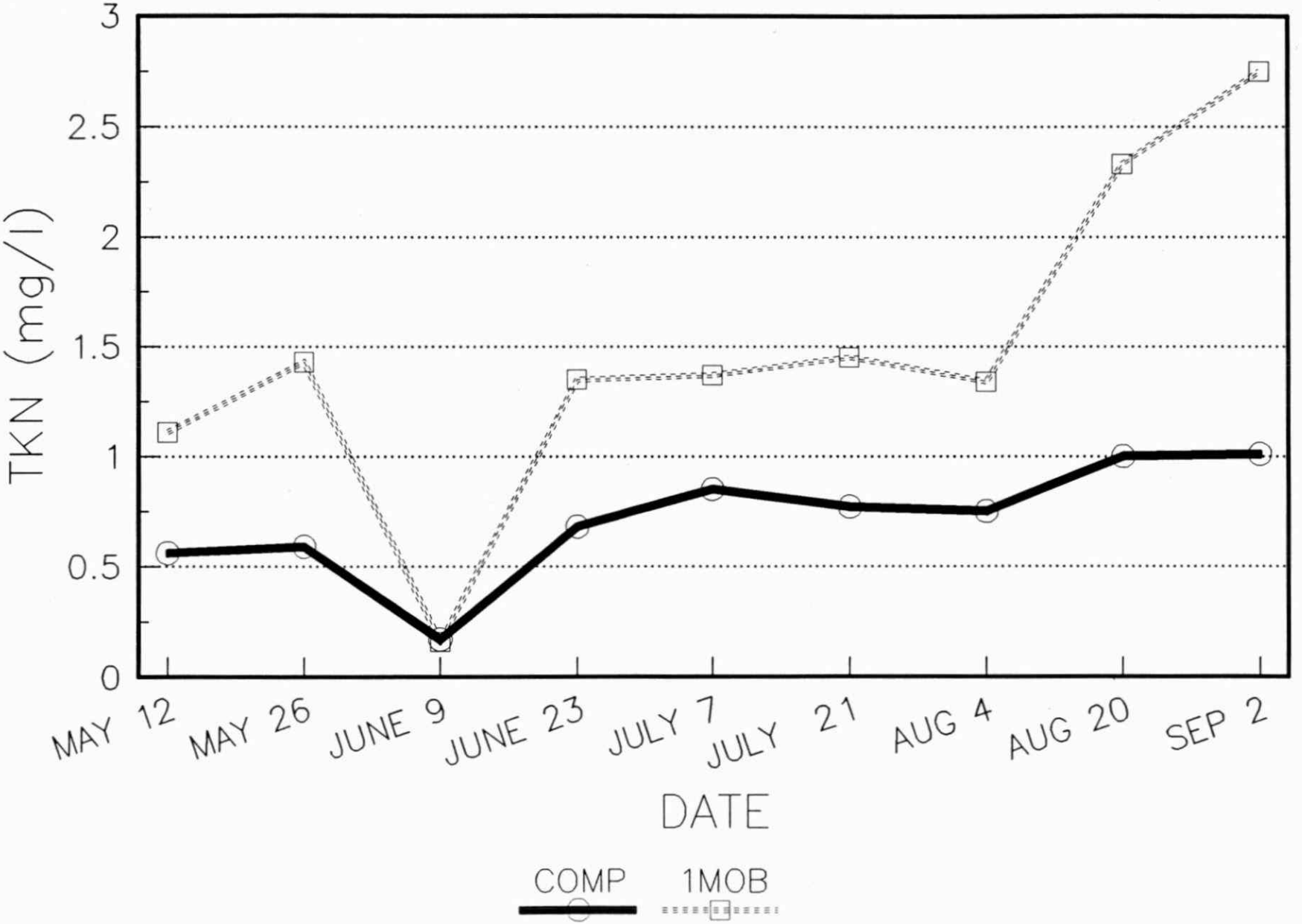

Figure 10: 1987 seasonal total nitrate nitrogen concentrations ($NO_2$ + $NO_3$) in the euphotic zone and at 1 MOB.

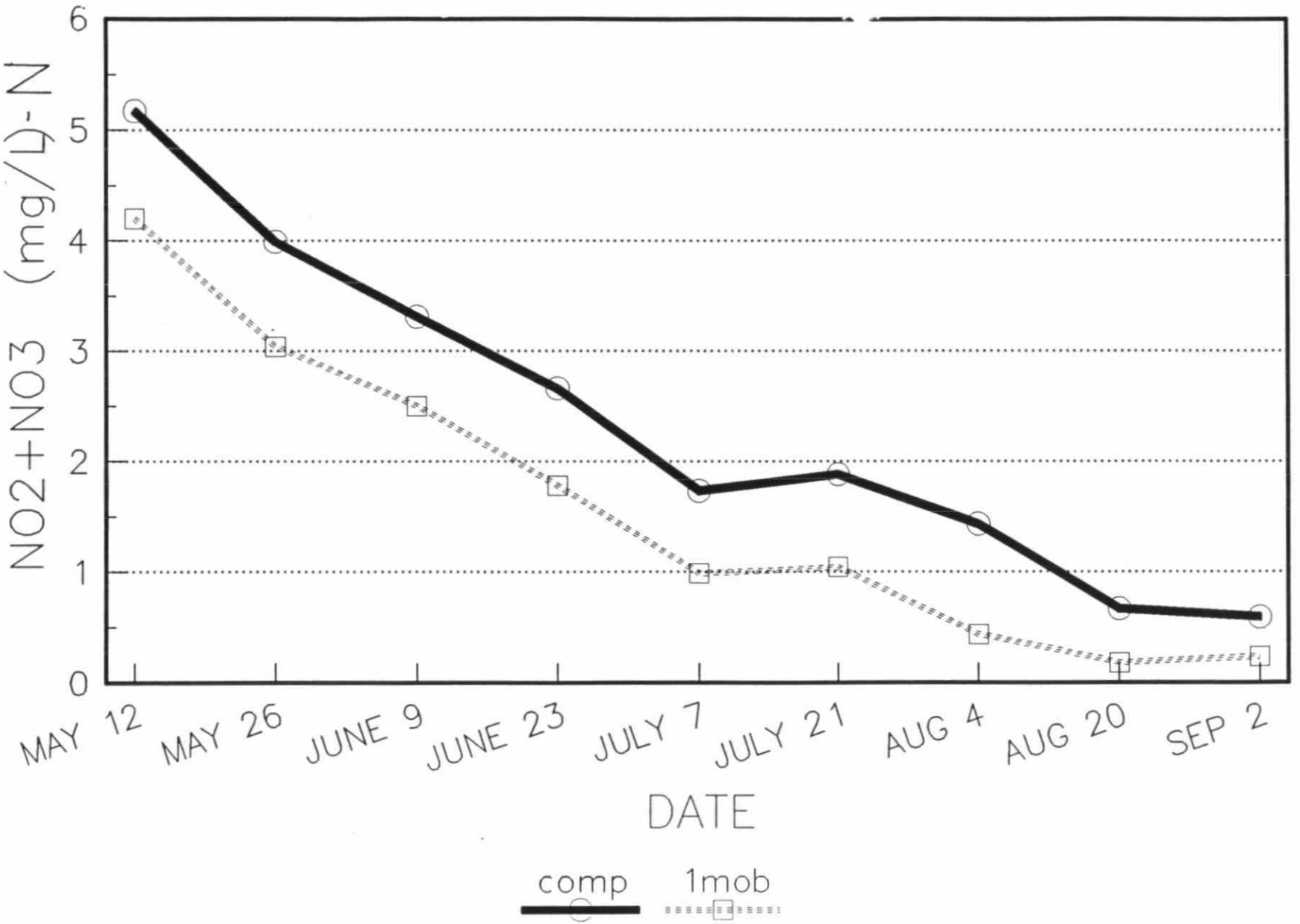

Figure 11: 1987 seasonal euphotic zone iron, magnesium and manganese concentrations .

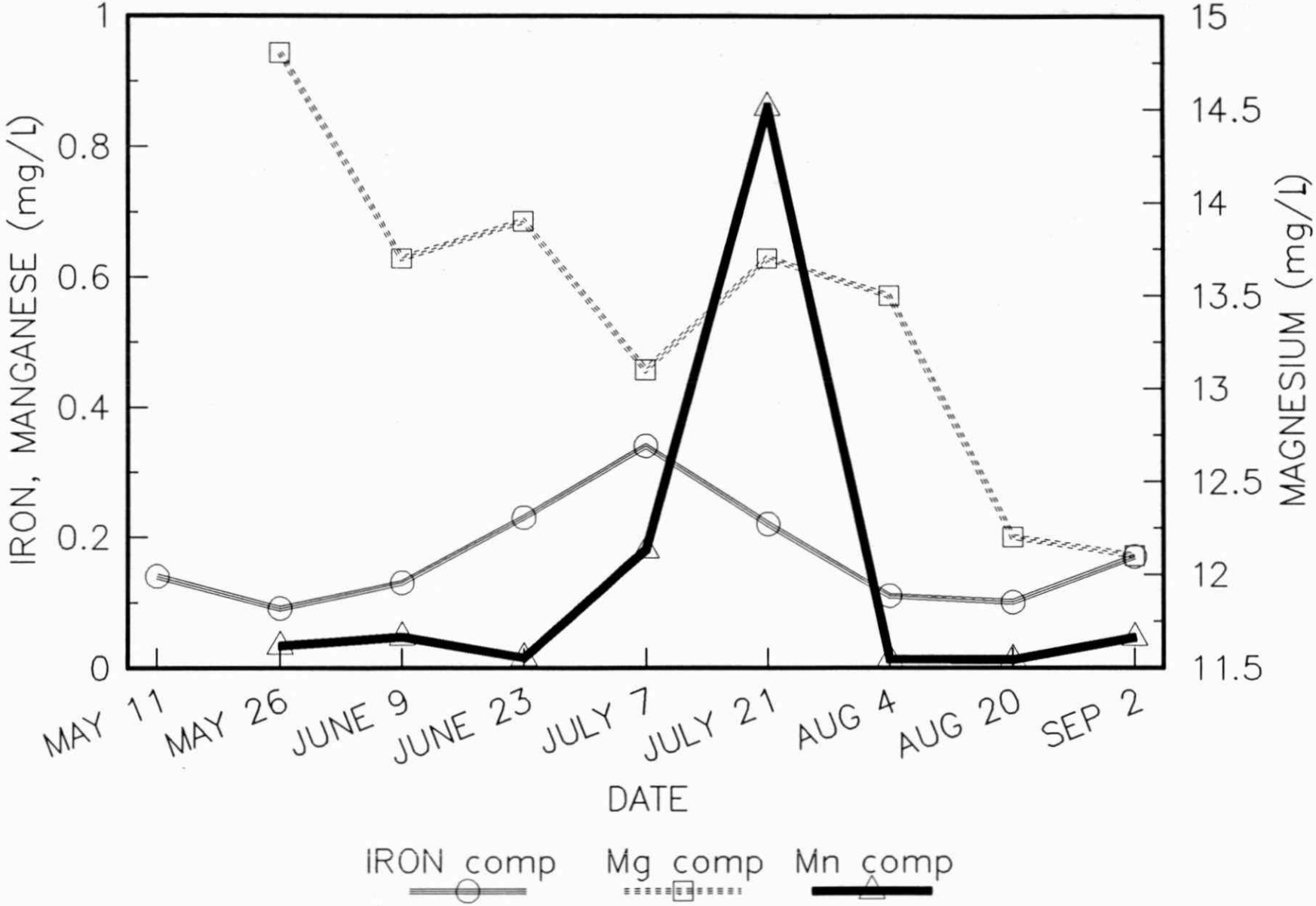

Figure 12: 1987 seasonal iron, magnesium and manganese concentrations at 1 MOB.

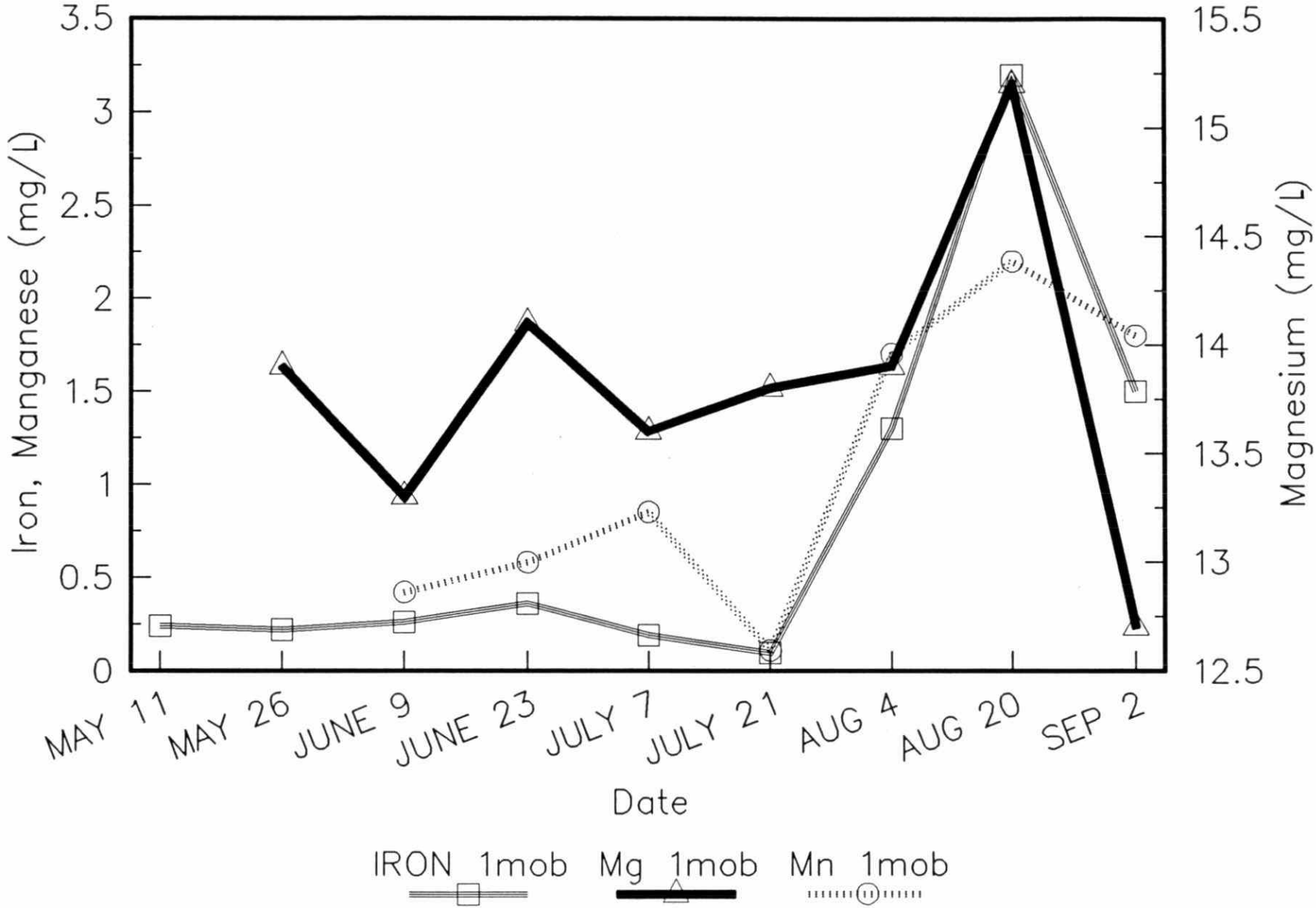

Figure 13: 1987 seasonal euphotic zone (comp) and 1 MOB conductivity readings .

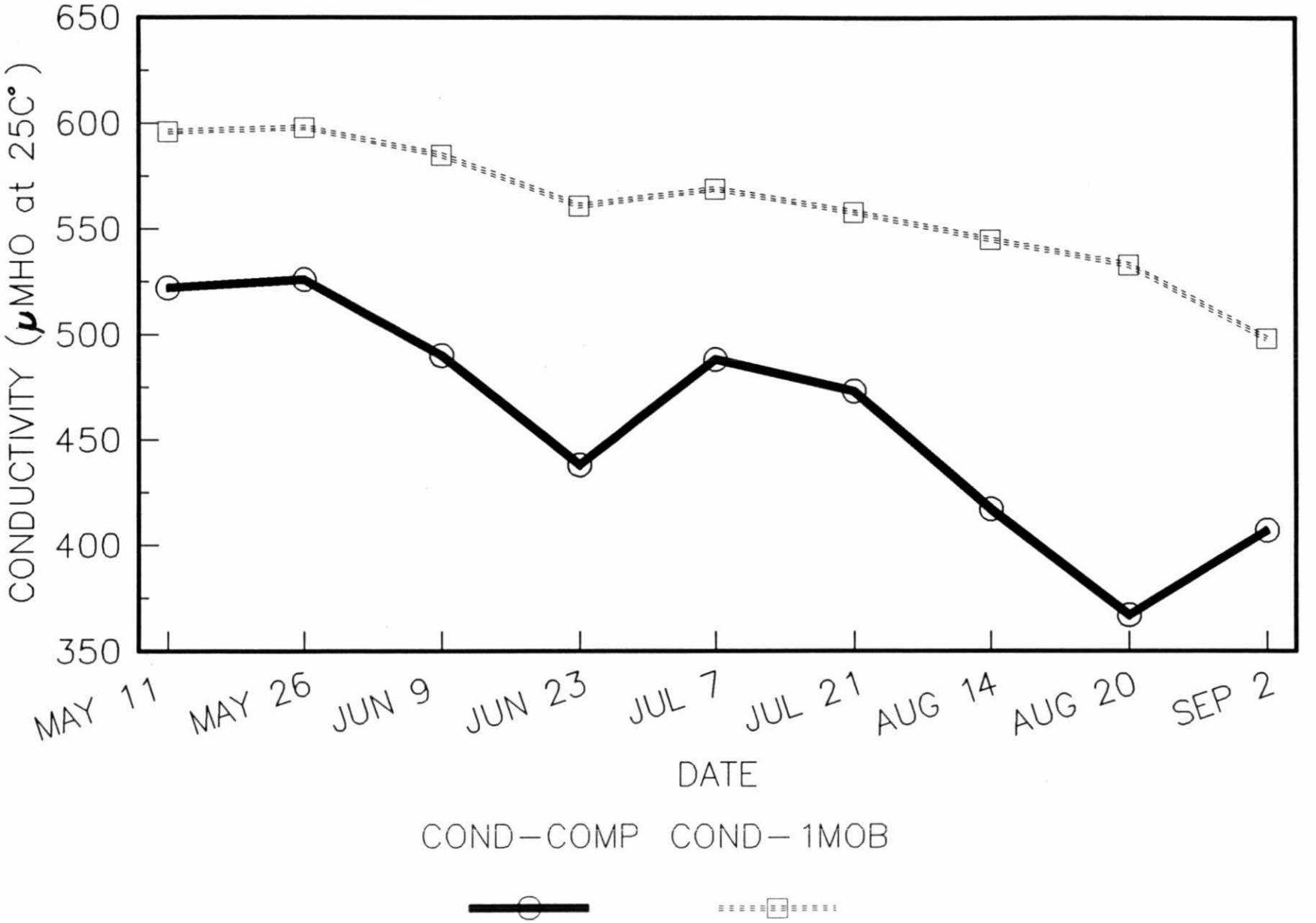

Figure 14: 1987 seasonal euphotic zone alkalinity, calcium DIC and silica levels.

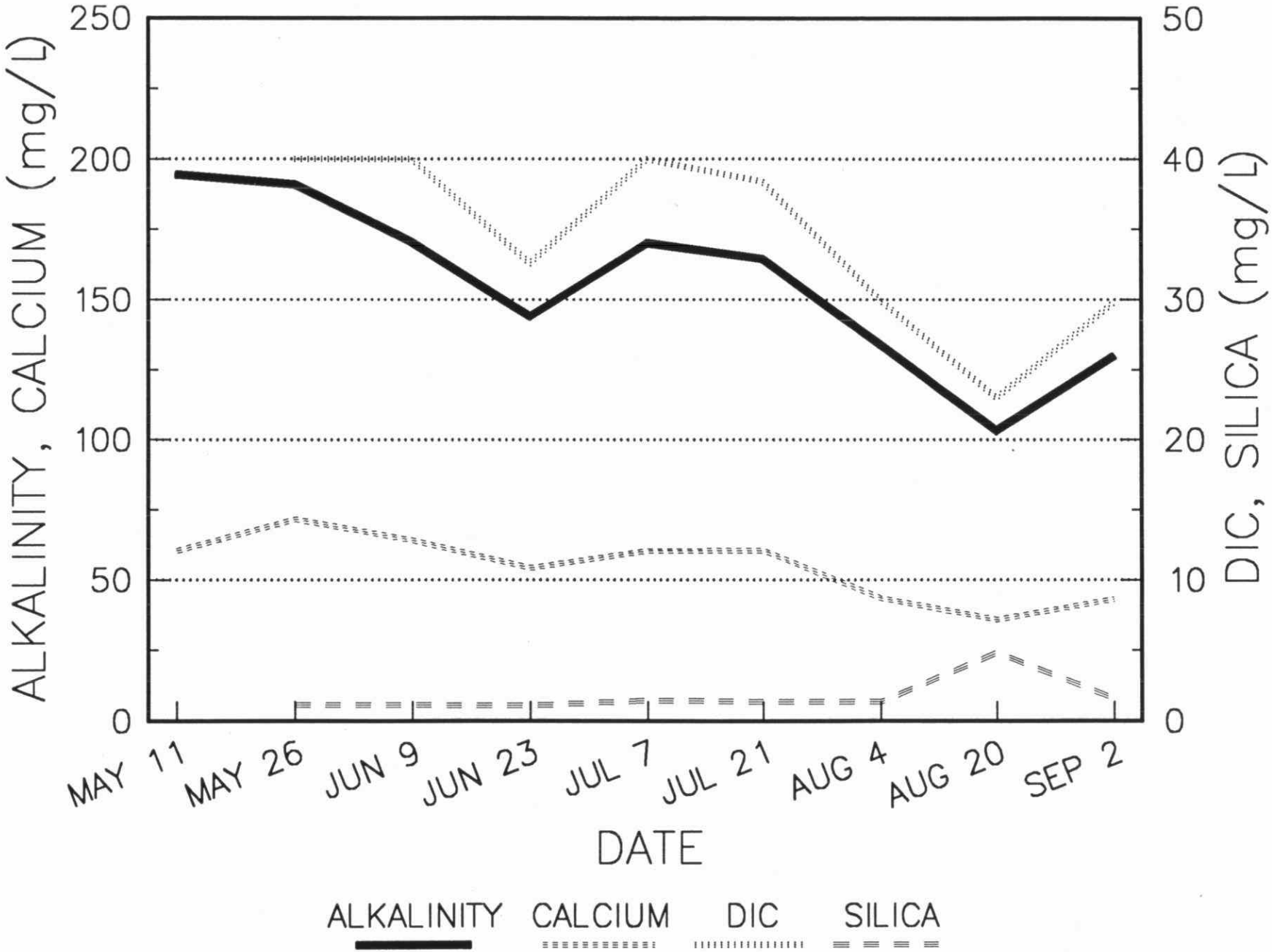

Figure 15: 1987 euphotic zone total chlorophyll a, and total nitrogen : total phosphorus ratios.

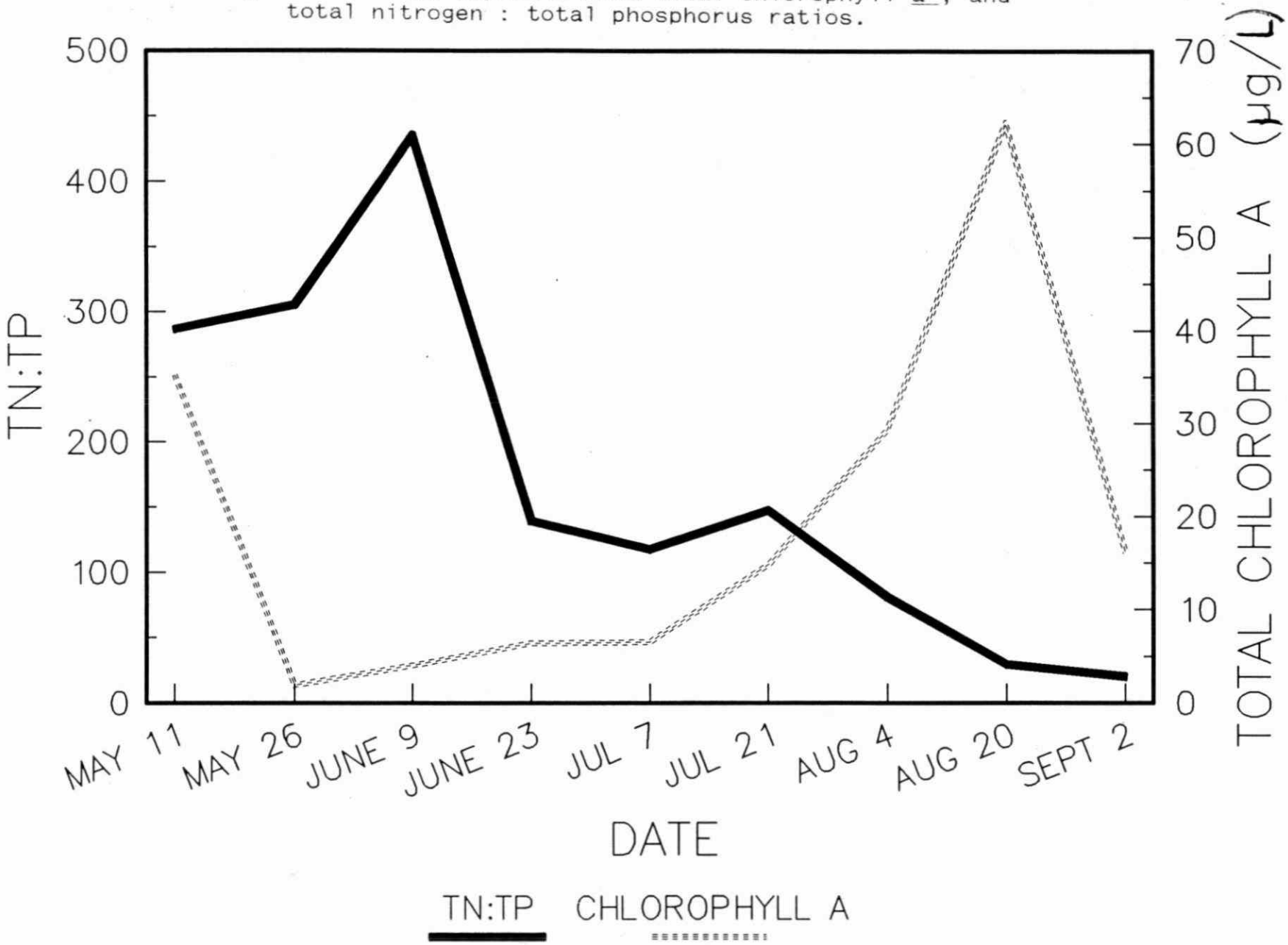

Figure 16: 1987 seasonal phytoplankton biovolume and total chlorophyll a concentrations .....

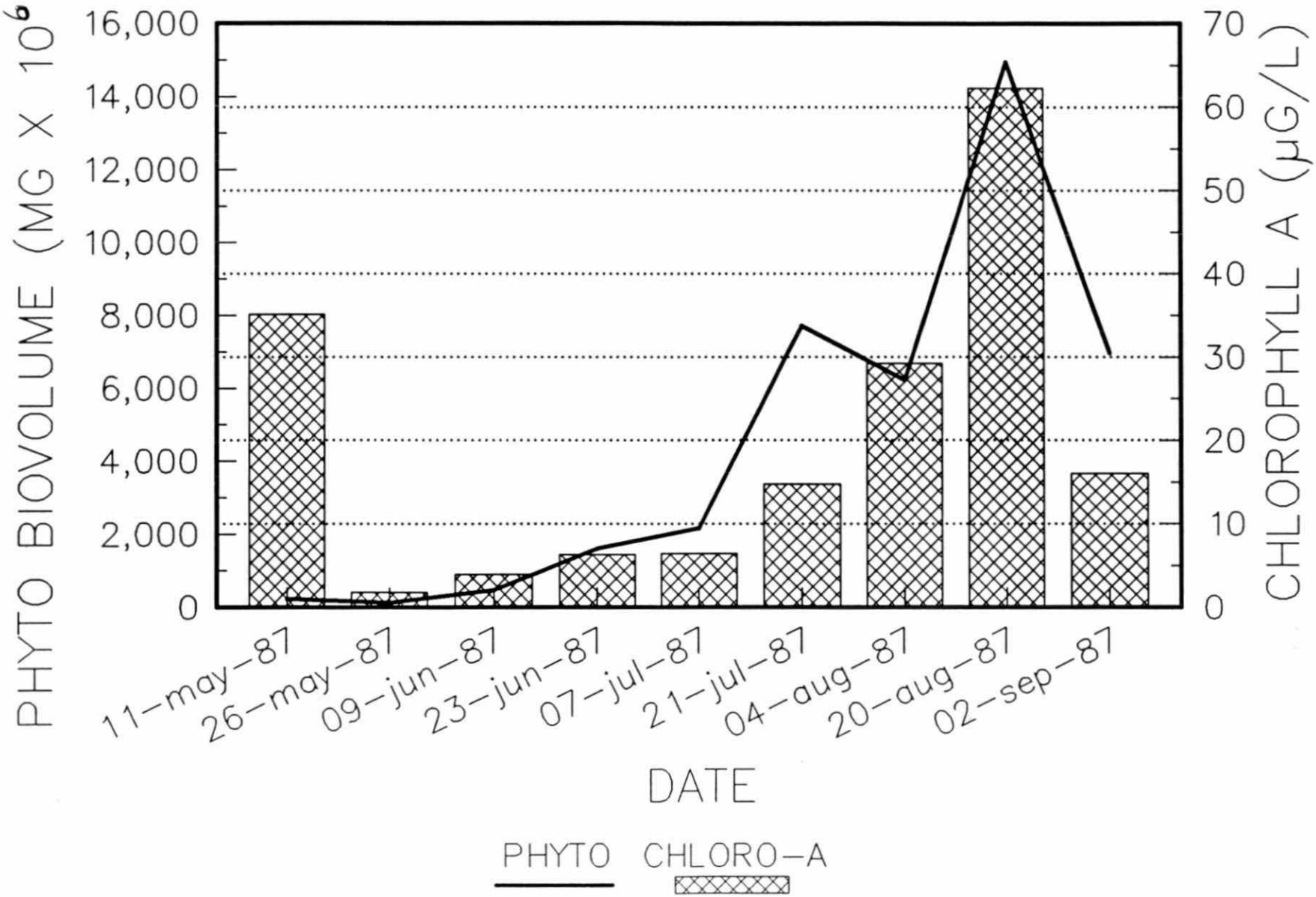

Figure 17: 1987 seasonal phytoplankton succession, as % composition of the total biovolume .

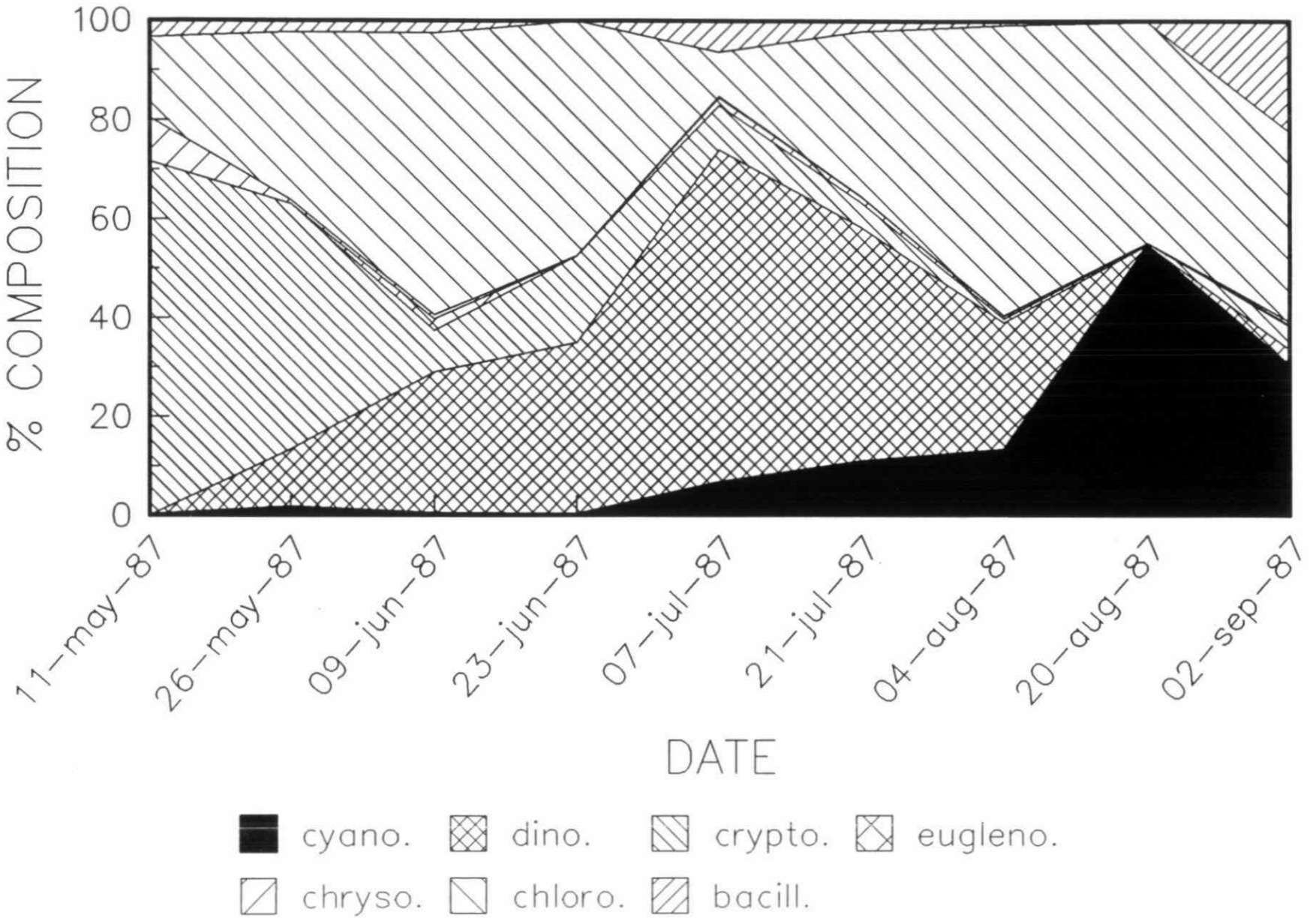

Figure 18: 1987 seasonal zooplankton biomass to phytoplankton biovolume relationship .

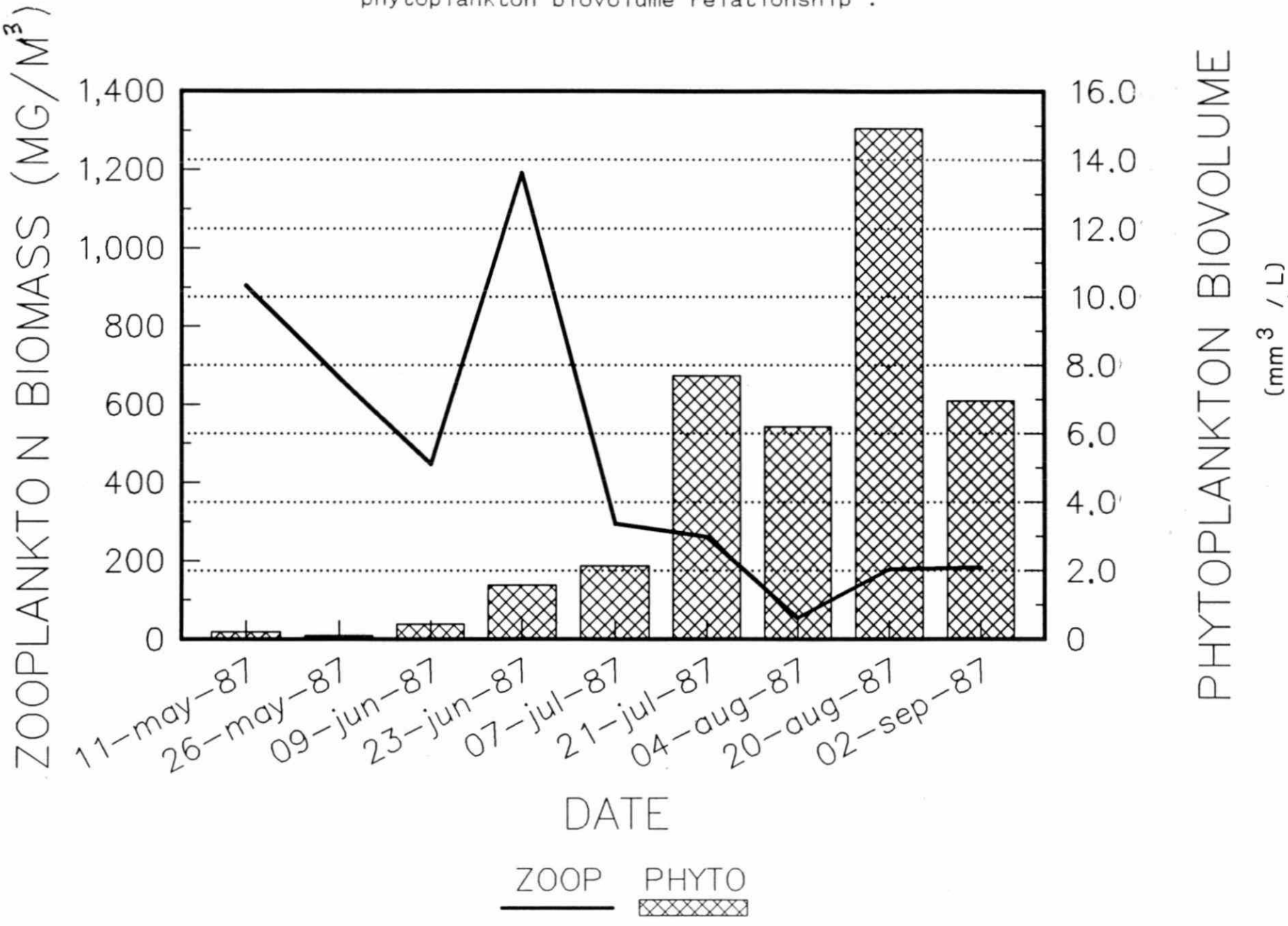

Figure 19: 1987 zooplankton succession as a % of total biomass.

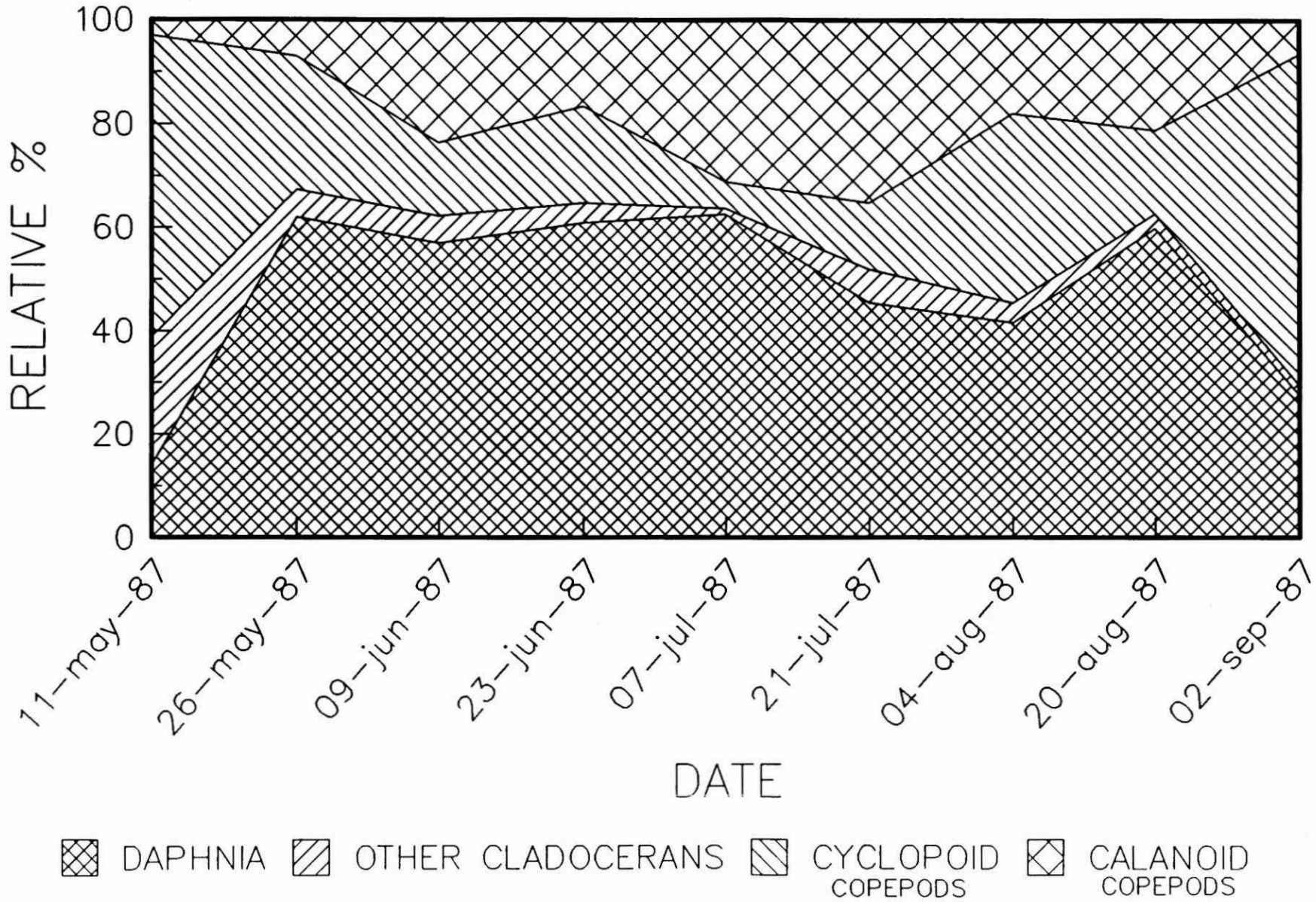

Table 1: Lisgar Lake: Mean annual ice-free season, euphotic zone concentrations 1987.

| | | LISGAR LAKE |
|---|---|---|
| SECCHI DISC | (m) | 2.2 |
| TOTAL CHLOROPHYLL A | (µg/L) | 19.6 |
| PHYTOPLANKTON | (mm /L) | 4.49 |
| % BLUE–GREENS | | 13 |
| ZOOPLANKTON | ($mg/m^3$) | 464 |
| TOTAL PHOSPHORUS | (µg/L) | 30 |
| PHOSPHATES | (µg/L) | 2 |
| NITRATES | (mg/L) | 2.38 |
| AMMONIUM | (mg/L) | .201 |
| KJELDAHL | (mg/L) | .709 |
| TN:TP | | 173 |
| pH | | 8.2 |
| CALCIUM | (mg/L) | 54.9 |
| HARDNESS | (mg/L) | 166 |
| CHLORIDE | (mg/L) | 31.8 |
| CONDUCTIVITY | (µmhos/cm) | 459 |
| TURBIDITY | (FTU) | 4.1 |
| DIC | (mg/L) | 34.2 |
| DOC | (mg/L) | 4.2 |
| IRON | (mg/L) | .16 |
| SILICA | (mg/L) | 1.7 |
| SULPHATES | (mg/L) | 30.5 |